1051-A-11
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NUREG/CR-2022
PNL-3772

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Technical Review of the Dispersion and Dose Models Used in the MILDOS Computer Program

Prepared by T. W. Horst, J. K. Soldat, T. J. Bander

Pacific Northwest Laboratory
Operated by
Battelle Memorial Institute

Prepared for
U.S. Nuclear Regulatory Commission

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Manuscript Completed: March 1982
Date Published: May 1982

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Prepared for
Division of Waste Management
Office of Nuclear Material Safety and Safeguards
U.S. Nuclear Regulatory Commission
Washington, D.C. 20555
NRC FIN B2215

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ABSTRACT

The MILDOS computer code is used to estimate impacts of radioactive emissions from uranium milling facilities. This report reviews the technical basis of the models used in the MILDOS computer code. The models were compared with state-of-the-art predictions, taking into account the intended uses of the MILDOS code. Several suggested modifications are presented and the technical basis for those changes are given.

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1.0 INTRODUCTION

The authors have reviewed the technical basis of the MILDOS computer code. The authors have attempted to take into account the intended uses of the code, but the individual user must still be responsible for judging the model structure and these review comments in the context of the proposed application. The portions which predict airborne and surface contaminant levels resulting from atmospheric diffusion, deposition and resuspension were reviewed by T. W. Horst and T. J. Bander. The portions concerned with food chains and human dosimetry were reviewed by J. K. Soldat. This review has been based on documents NUREG/CR-0553, NUREG-0511[a] and Regulatory Guide RH 802-4.[b]

The MILDOS models for diffusion and deposition are adequate applications of the state-of-the-art to the prediction of environmental impact. Several simple improvements to these models are discussed in Sections 2.1 and 2.2. In contrast, the model for resuspension is based on limited data and does not properly describe the spatial distribution of resuspended contamination. An alternate method, which predicts an upper limit for the airborne and surface contamination, is suggested in Section 2.4.

The MILDOS food chain and human dosimetry models were reviewed to determine if the models would yield results which were reasonable considering the state-of-the-art and the intended uses of the MILDOS code. The accuracy of any model, especially one involving environmental processes, is difficult to determine without field data for comparison with model predictions. Lacking such data, judgments on the reasonableness of the assumptions implicit in the equations, the parameter values selected, and the consistency of the approach were based on current practice in the field. Several recommendations are made for these models in Section 3.3.

(a) NUREG-0511 is a draft version of NUREG-0706, The Generic Environmental Impact Statement on Uranium Milling, September 1980.

(b) RH 802-4 is a draft NRC Regulatory Guide which is in the process of being revised for final issue.

2.0 DISPERSION MODELS

2.1 DIFFUSION

The atmospheric diffusion is estimated with a standard Gaussian diffusion model, accounting for a finite atmospheric mixing depth, plume rise at the source and the possibility of distributed area sources.

2.1.1 Diffusion Parameters

The Gaussian diffusion model describes σ_z, the vertical spread of airborne contamination, with a set of empirical relations which depend on atmospheric stability and distance from the source. Near the source these relations are linear with distance. NUREG/CR-0553 states in Section 2.2 that this is unreasonable for distances less than 100 m and sets σ_z at those distances equal to its value at 100 m, i.e.

$$\sigma_z(r) = \sigma_z\ (100\ m),\ r \leq 100\ m. \qquad (2\text{-}1)$$

This is certainly not correct for point sources, such as stacks, and for non-point sources no basis has been given for the choice of 100 m in Equation (2-1). NUREG-0511 states more correctly in Appendix G that the Gaussian plume model may not be accurate in the immediate vicinity of the source, presumably due to some initial mixing of the contaminant, and declines to calculate air concentrations for distances less than 100 m from the source. Both approaches, however, will incorrectly account for dry deposition from ground-level sources since this is greatest near the source. Further discussion on deposition is found in a later section of this review.

2.1.2 Mixing Depth

For neutral and unstable atmospheric conditions, the well-mixed surface layer is capped by an elevated inversion (stable layer) which limits the vertical mixing to a depth L. Hence, at large downwind distances, MILDOS imposes an upper limit L on the vertical mixing. For stable conditions the base of

the stable layer is at the surface. Diffusion within that layer is characterized by a σ_z which grows slowly with downwind distance and itself reaches a constant value at large downwind distance. Hence, MILDOS imposes the limit L only for unstable and neutral conditions, Pasquill classes A through D (NUREG-CR-0553, Section 2.3, incorrectly states A and D), and not for stable conditions, classes E and F.

Observations of the mixing depth are routinely available only twice per day, 0000 GMT (Greenwich Median Time) and 1200 GMT. For simplicity, MILDOS averages these morning and afternoon observations to get a single annual-average mixing depth. Two choices are given in Section 2.4 of NUREG/CR-0553 for this averaging. The most reasonable of these is the average value of 1/L since the computed airborne concentrations are inversely proportional to L.

2.1.3 Direct Air Concentrations

The annual-average concentration of contaminant is calculated by integrating the Gaussian diffusion model in the crosswind direction. This integrated value is then redistributed in a triangular form with the peak along the downwind direction, considered as the centerline of the plume. The concentration decreases linearly to zero at 22-1/2° (one sector width) in either direction from the centerline of the wind direction sector. Distributing in this manner smooths out the annual average concentration calculated around a circle at some given distance from a source.

The vertical spread of the contaminant is determined from the empirical σ_z's, which for stable atmospheric conditions are used at all downwind distances. However, for unstable and neutral conditions a determination is made of the distance x_L at which the vertical dispersion becomes greater than 0.47 times the mixing depth. For all distances between x_L and $2x_L$, the concentration is then determined by taking a linear interpolation between the concentrations at x_L and $2x_L$, where the concentration at $2x_L$ is calculated assuming the vertical dispersion to be uniform throughout the mixing depth. Thus for unstable and neutral stabilities, the empirical σ_z's are not used at distances greater than x_L. The plume depletion integrals, however, use σ_z between x_L and $2x_L$ for all stabilities and thus are not consistent with the diffusion formulation.

2.1.4 Plume Rise

MILDOS accounts for plume rise above the height of the source with the formula

$$\Delta h = 1.5 \frac{wD}{u} \qquad (2\text{-}2)$$

where w is the efflux velocity, D is the stack diameter and u is the wind speed. Recent observations have shown that plume rise also depends on downwind distance r, and Briggs (1969, 1975) recommends

$$\Delta h = 3\left[\frac{wD}{2u(1 + 3u/w)}\right]^{2/3} r^{1/3} \qquad (2\text{-}3)$$

with a maximum value of

$$\Delta h = 3 \frac{wD}{u} \qquad (2\text{-}3a)$$

Figure 2.1 compares the predictions of Equations (2-2) and (2-3). Equation (2-3) is no more difficult to include in MILDOS than Equation (2-2) and for most distances it increases Δh by a factor of two. Note that Equations (2-2) and (2-3) are applicable to uranium mill stacks for which thermal buoyancy can be neglected. In cases where this is not true, the additional plume rise due to buoyancy must be included.

2.1.5 Area Sources

MILDOS accounts for the finite horizontal extent of area sources by subdividing them into squares of width d. For the horizontal spread of the plume, each subdivision is replaced with a virtual point source located a distance $r = 8d/\pi$ (based on 22.5° wind direction sectors) upwind of the subdivision's center. For vertical spread and to account for deposition, a virtual source is located at the center of the subdivision. A comparison with an exact solution obtained by numerical integration (Horst, 1978) shows that if d is less than 300 meters this latter approximation is correct within a factor of two.

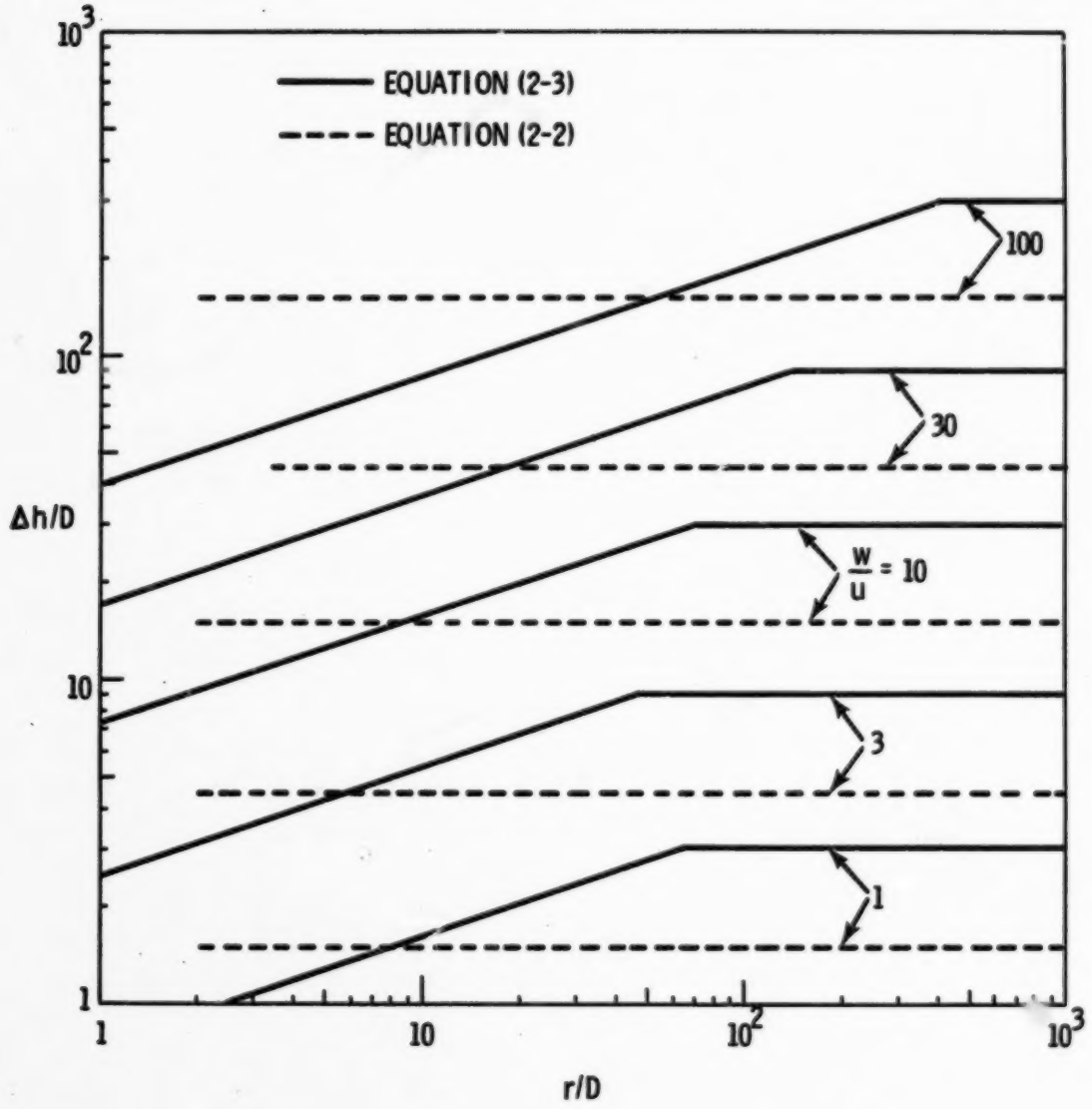

FIGURE 2.1. A Comparison of Plume Rise Formulas

2.2 DEPOSITION

MILDOS models the deposition flux as a deposition velocity v_d multiplied by the ground-level airborne contamination C(z=0). The contaminant plume is depleted by the source depletion method. These are acceptable state-of-the-art techniques for environmental impact applications. However, the inclusion of resuspension would allow plume depletion to be neglected entirely, a major simplification. It is also recommended that state-of-the-art techniques be used to estimate deposition velocities as a function of wind speed, particle size, and surface characteristics.

2.2.1 Deposition Velocity

The MILDOS estimation of deposition can be made more realistic by using current models to account for the dependence of deposition velocities on particle size, wind speed and surface roughness. The empirical model of Sehmel and Hodgson (Sehmel, 1980a) shows a minimum deposition velocity of much less than 1 cm/sec for particles between 0.1 and 1 μm diameter and an increase in deposition velocity roughly proportional to particle diameter for smaller particles and to the square of particle diameter for larger particles. Deposition velocity also increases with the wind speed and this correlation should be explicitly included in the computation of the annual-average deposition. Finally, deposition velocities also increase with surface roughness. Although the dependence is weaker than for particle size or wind speed, it may also be accounted for with Sehmel and Hodgson's model.

The MILDOS deposition model would also be improved by a finer subdivision of particle sizes, but with the drawback of additional computational cost.

2.2.2 Plume Depletion

MILDOS uses the source depletion model to account for the loss of material from the airborne plume by dry deposition. However, MILDOS sets the lower limit of the depletion integral, F_1 in NUREG/CR-0553, equal to a downwind distance of 100 m. This, and the redefinition of σ_z in Equation (2-1), may be motivated by the fact that the integral is undefined for a ground-level source if the lower limit is properly set to zero and the deposition is parameterized with C(z=0).

For elevated sources the integral is well defined, however, and should begin at the source. For ground-level sources the depletion integral is well defined only if deposition is parameterized with $C(z_*)$ where $z_* \neq 0$, but unfortunately the integral becomes a sensitive function of z_*.

These plume depletion considerations may be academic, however. It is recommended in the following section that plume depletion be neglected entirely in order to conservatively account for resuspension. The computation of airborne contamination, deposition, and surface contamination based on the undepleted airborne plume will account for deposition from the original plume, as well as for all subsequent resuspension and redeposition.

2.3 RESUSPENSION

Air concentrations of resuspended contamination are predicted in MILDOS by multiplying the surface contamination G by a resuspension factor K. The surface contamination is calculated only from the deposition of airborne contamination transported directly from the original source. The resuspension factor is assumed to be spatially uniform and to decay exponentially with time at a "weathering" rate λ_k. This method of accounting for resuspension is based on very limited data and does not properly describe the spatial distribution of resuspended contamination. It is best applied, if at all, only in the immediate vicinity of the maximum surface contamination. An alternative method is suggested which is conservative and is commensurate with our present knowledge of resuspension.

2.3.1 Resuspension Factor

Measurements of resuspended airborne contamination are currently limited to a very small number of circumstances, notably the arid environment of nuclear test sites. Even for these particular environments (which are similar to those of many existing uranium mills) the resuspension parameters have a large uncertainty associated with them. Observed values of K for resuspension by the wind range from 10^{-10} m^{-1} to 3×10^{-4} m^{-1} and, since they vary by several orders of magnitude even within a single set of observations, taking an average resuspension factor cannot be justified (Sehmel, 1980b). Estimates of

λ_k vary from 7.2 yr^{-1} to 0.68 yr^{-1}, but in controlled experiments Sehmel (1980b) has also found no reduction of the resuspension with time, i.e. $\lambda_k = 0$. Lassey (1980) and Kocher (1980) have recently discussed these uncertainties and note that as a consequence it is not even possible to determine the importance of resuspended contamination relative to the airborne contamination transported directly from the original source.

A second weakness of the resuspension factor is that it does not properly account for the dependence of the resuspended contamination on the spatial distribution of surface contamination or for the dependence on source-receptor separation, because it relates airborne contamination to the local surface contamination, rather than to the upwind distribution of contamination (Healy, 1977). Observations of the dependence on upwind, rather than local, surface contamination were noted by Stewart (1967). Horst (1977) demonstrates that for uniform surface contamination the ratio of resuspended contamination to surface contamination increases with distance from the upwind edge of the contaminated area. For non-uniformly contaminated areas, this ratio increases slowly as the peak contamination is approached from the upwind side but increases very rapidly with downwind distance from the peak. Thus, in most cases, a uniform resuspension factor will not predict the correct spatial distribution of resuspended contamination.

2.3.2 Resuspension Rate

The latter weakness of the resuspension factor can be illustrated by examining the predictions of a more realistic model which assumes that the vertical resuspension flux is proportional to the local surface contamination through a resuspension rate Λ. This model calculates the resuspended contamination by accounting for atmospheric transport and diffusion between the receptor and the upwind distribution of surface contamination (Horst, 1977; Horst, 1979).

The predictions of this model are shown schematically in Figure 2.2 for the situation modeled by MILDOS, where a localized source produces annual-average, airborne contamination C at breathing level. There are no scales on the axes of Figure 2.2 because its purpose is to show the relationships among

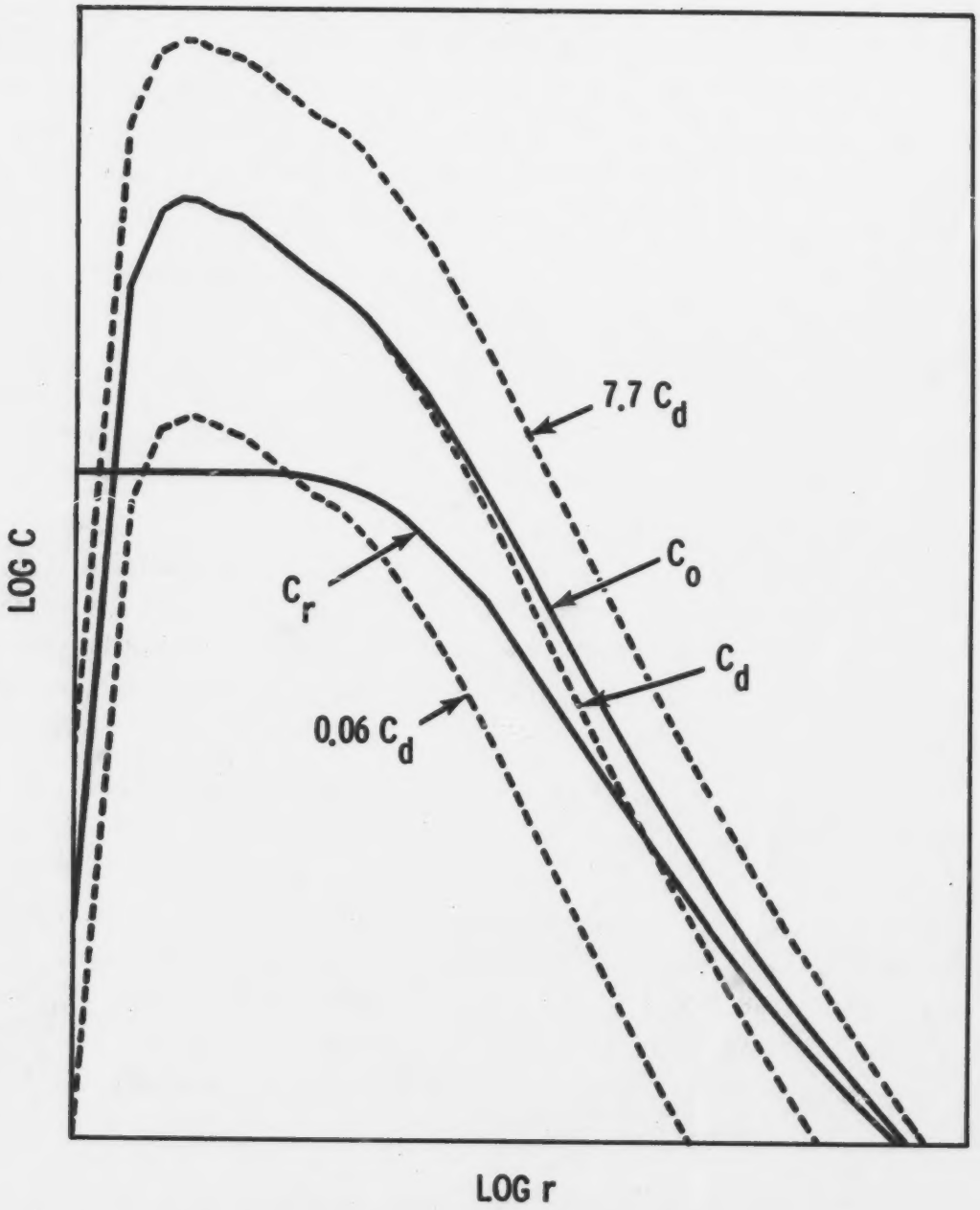

FIGURE 2.2. Annual-Average Airborne Contamination at Breathing Height as a Function of Distance from the Original Source. The resuspended contamination is predicted to be C_r by the resuspension rate model, $0.06C_d$ and $7.7C_d$ by resuspension factor models.

the various predictions of airborne contamination; actual magnitudes of the curves depend on the diffusion climatology of the site, source height, deposition velocity, resuspension rate, etc. The various C are averaged over all wind directions and hence are functions only of the distance r from the source. With no deposition or resuspension the airborne contamination is C_o; with deposition the direct airborne contamination is reduced to C_d.

The surface is contaminated by deposition both from the direct airborne contamination and from the resuspended contamination C_r. If all of the surface contamination is assumed to be available for resuspension, i.e., neglecting losses by weathering, the resuspended contamination is maximized. The surface contamination increases with time until resuspension balances deposition and $C_d+C_r \approx C_o$. More exactly, the resuspended contamination is found to be less than C_o-C_d, the deposition loss from the direct airborne contamination (Horst, 1979). An exception to this conclusion may be seen in the immediate vicinity of an elevated source, where the direct airborne contamination does not reach the surface. In this case the resuspended contamination is sustained by surface contamination which has been deposited in the past on the opposite side of the source.

The relative magnitudes of the resuspended contamination as predicted by the resuspension rate model and by MILDOS depend on the particular values of K, λ_k and deposition velocity v_d. MILDOS, as well as other resuspension factor models, predicts the resuspended contamination to be proportional to C_d. Kocher (1980) has calculated the proportionality factor for several models and his results, which range from 0.06 to 7.7, are included in Figure 2.2. For particles with a deposition velocity of 1 cm sec^{-1}, the default parameters in MILDOS predict an intermediate value of 0.63. A comparison of these values with C_r predicted by the resuspension rate model shows that a resuspension factor model will agree with a resuspension rate model at no more than two distances from the source. This follows from the relative shapes of the curves for C_d and C_r, which are determined by the diffusion-deposition process and are independent of the resuspension parameters. MILDOS cannot reproduce both $C_r \ll C_o \sim C_d$ near the source and $C_r \sim C_o \gg C_d$ far from the source.

2.4 CONCLUSIONS

Without resuspension MILDOS predicts the airborne contamination to be C_d, the direct airborne contamination depleted by dry deposition. If the model estimates for the source strength, atmospheric diffusion and dry deposition are all correct, this would underestimate the airborne contamination. Accounting for resuspension by the resuspension factor model gives a more conservative estimate for the total airborne contamination $(1+R)C_d$. MILDOS predicts R = 0.63. However, the complexity of the resuspension process and the lack of observational data makes it very difficult to reliably estimate K, λ_k, and R. Further, it has been shown that the resuspension factor model predicts the wrong dependence of resuspended contamination on distance from the source. It is best applied, if at all, only in the immediate vicinity of the maximum surface contamination, where the local surface contamination may be more important than upwind contamination in determining resuspended airborne contamination.

The resuspension rate model is more realistic, but unfortunately is similarly indeterminate, again due to a lack of data on resuspension and weathering rates. In contrast to the resuspension factor model, however, the resuspension rate model establishes a physical upper limit for C_r, i.e., $C_r \leq C_o - C_d$. Even though this is an estimate, it is known to be conservative and may in many cases be less than resuspension factor predictions. Thus, it is recommended that the annual-average airborne contamination, C_d+C_r, be estimated for a continuous source simply by neglecting dry deposition from the direct airborne contamination. This estimate must be modified to account for resuspension immediately below an elevated source. Here the airborne contamination can be conservatively estimated to be equal to the peak value of C_o. For a source of finite duration, the total exposure (or time-integrated concentration) may be estimated in a directly analogous manner.

These estimates assume that the surface contamination increases by dry deposition, from both the direct and the resuspended airborne contamination, until resuspension balances deposition and $C_d+C_r \approx C_o$. Thus the surface contamination is given by Equations (4.34) and (4.35) of NUREG/CR-0553, but with the undepleted direct concentration C_o instead of C_d and the resuspension rate Λ instead of the "denudation coefficient" μ or "environmental loss constant" λ_e

(RH 802-4). Unfortunately there are few measurements of Λ and these vary from 10^{-5} yr^{-1} to 10^{3} yr^{-1} (Sehmel, 1980b). RH 802-4 and NUREG-0511 assign a value of 0.014 yr^{-1} to λ_e, but do not cite any data to support that value. In view of this large uncertainty, an alternate approach would take no credit for these losses, i.e. $\Lambda = \mu = \lambda_e = 0$.

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3.0 FOOD CHAIN AND HUMAN DOSIMETRY MODELS

3.1 FOOD CHAINS

This section of the review is based principally on Regulatory Guide RH-802-4 (USNRC, 1979) and, to a limited extent, NUREG-CR-0533 (Momeni, et al., 1979). The food chain model is based on equations in USNRC Regulatory Guide 1.109 (USNRC, 1977) which were patterned after the code FOOD (Napier, et al., 1980)[(a)]. The models in Regulatory Guide 1.109 are, however, different from FOOD in that certain parameters were either omitted or changed in value; e.g., the fractional retention of deposited airborne radionuclides by vegetation, and the translocation of deposited materials from vegetation surfaces to portions of the plants consumed. Ingestion dose factors were obtained, in the most part, from NUREG-0172 (Hoenes and Soldat, 1977).

3.1.1 Retention

The fraction of deposited airborne radionuclides retained by vegetation was given as 0.25 in FOOD (Napier, et al., 1980) and HERMES (Fletcher and Dotson, 1971) based on data for particulates and iodine found in the literature. For convenience this same value of 0.25 was assumed for sprinkler application of contaminated water, since no data were found in the literature for this parameter. In Regulatory Guide 1.109, however, the retention factor was raised to 1.0 for iodine, lowered to 0.2 for other particulates, and left at 0.25 for sprinkler irrigation.

The MILDOS code uses the value of 0.2, from Regulatory Guide 1.109, for retention of the particles containing uranium and daughters. This value is acceptable considering the small amount of data available. In fact, it may be preferable to a value of 0.25 as the data only warrant one significant figure.

3.1.2 Translocation

Translocation is the term adopted in HERMES for the transfer of deposited radionuclides from the external surfaces of vegetation to the portion actually consumed by man or animal.

(a) FOOD, in turn, is a simplified version of the terrestrial food chain and human dosimetry portions of the code HERMES (Fletcher and Dotson, 1971)

The values of the parameter actually refer to the ratio of concentration of a radionuclide in the edible portion divided by the average concentration obtained by dividing the total activity of the externally deposited nuclide by the total mass of the vegetation. Limited data on certain specific nuclides in grains and potatoes obtained from the literature are summarized in the HERMES report (Fletcher and Datson, 1971). It was assumed then that the data for grain could be used for all above-ground vegetables, and that the values for potatoes also applied to root vegetables. The values of the translocation factors ranged from 0 up to 0.1 for both categories of vegetation. As a simplification, the FOOD code was designed to use a translocation factor of 0.1, the maximum found in the literature for all nuclides for both categories.

Because it was assumed that the portions of green leafy vegetables and pasture grass eaten would include the outer surfaces, the value of the translocation factor was taken as 1.0 for that category for all nuclides in both the HERMES and FOOD codes. Regulatory Guide 1.109 eliminated the translocation factor with the result that all parts of the plant would be uniformly contaminated. In attempting to restore the translocation factor in MILDOS, the value of 0.1 was applied to all root crops and a value of 1.0 to all above-ground plants.

If, as assumed, the concentration ratios (plant/soil) in the literature are derived directly for the portion consumed (such as grains) then there is no need for an additional translocation factor for the root pathway. For this reason, none has been applied in any of the terrestrial food codes discussed here.

I recommend that the translocation factor in MILDOS be modified to a value of 1.0 for all leafy vegetables and pasture and hay, and a value of 0.1 for all other vegetation including grain, above-ground vegetables and root crops (Fletcher and Dotson, 1971). These values should be used for all nuclides. Although no data were given in Fletcher and Dotson (1971) for translocation of uranium and daughter products, it seems logical to assume that for these nuclides the value of the translocation factor would not exceed the maximum of 0.1 listed there and adopted for all nuclides in the FOOD code.

Other equations and parameter values in the terrestrial food chain portion of MILDOS, such as the soil areal density of 240 kg m^{-2} and the 14-day weathering loss half-time for radionuclides on vegetation, reflect current usage.

3.2 HUMAN DOSIMETRY

MILDOS calculates radiation doses for all of the pertinent pathways covered by the transport and food chain portions of the code using dose conversion factors.

3.2.1 Categories of Dose Calculated

There are several different categories of radiation dose that can be calculated for either the individuals or the populations involved. The specific types of dose calculated should include one that matches as closely as possible those addressed in any applicable radiation dose standards. For radionuclides, most guides speak to annual dose or dose rate; i.e., mrem yr^{-1} or rem yr^{-1} to various specific organs or total body. Limits are seldom promulgated for collective doses, numerical limits (i.e., man-rem) are seldom expressed, but rather the ALARA (as low as reasonably achievable) principle is invoked.

Standards usually give very little guidance on the proper method of calculating doses used to determine compliance with annual dose limits. As a result, several different types of doses are being calculated today and the exact differences between the methodologies are not always clear. Listed below are four types of doses often calculated. Many other types are possible.

1. The first year dose from one year's exposure and uptake
2. The 50-year (or 70-year) committed dose from one year's exposure and uptake.
3. The integrated dose from all the exposure and intake which occurs over a 50-year (or 70-year) exposure period.
4. Calculation of the probable maximum annual dose by compounding each prior year's contribution to the current year's body burden of radionuclides, and doing this for each year throughout a 50-year (or 70-year) exposure period.

The first method is useful for an annual assessment of the management of radioactive effluents for comparison with prior years' dose where the effluent release rates to the environment may vary over time. The second method

is the one currently employed in MILDOS and Regulatory Guide 1.109 for calculation of radiation doses to individuals. The third is the logical method for calculation of realistic total collective doses for conversion to health effects. It can also be used for calculating total accumulated individual doses. The fourth is designed to yield values comparable to annual dose limits stated in terms of dose yr^{-1}.

The practice of using dose factors which convert an annual intake of a radionuclide into a 50-year committed dose for comparison with annual limits has become widespread. Most guides, however, do not mention such a procedure as an acceptable alternative to method 4.

The calculated 50-year committed dose and an annual limit would be comparable only if the intake rate of the radionuclide were constant for 50 years. Then the body burden accumulated after 50 years would deliver a dose rate (in units of rem yr^{-1}) of the same numerical value as the 50-year committed dose (in units of rem per 50 years).

Barring constant intake, the 50-year dose commitment calculated must be made for the year of highest intake to ensure that the maximum annual dose in any year does not exceed the guidelines. This normally corresponds to the year of highest concentration in environmental media (air, soil, water, and food). MILDOS allows this option.

There are two situations where this type of dose calculation is inappropriate.

1. The effluent release period or the exposure period is less than 50 years.
2. The peak concentrations in different media occur at widely separated times.

Instances of relatively constant environmental concentrations are indeed present for uranium tailings piles. These include radon releases from the piles during periods when management practices are constant.

3.2.2 Inhalation Dose

MILDOS calculates inhalation doses for the lung as weighted averages over the nasopharyngeal (N-P), tracheobronchial (T-B), lymph (L), and pulmonary (P)

regions defined by the Task Group Lung Model (TGLM) of the International Commission on Radiological Protection (ICRP, 1966). Up until recently the term "lung dose" usually referred to the dose to the pulmonary compartment of the respiratory tract. This convention was applied with both the older lung model in ICRP Publication 2 (ICRP-59) and the TGLM. ICRP Publications 26 and 30 (ICRP-77, ICRP-78) state that three regions of the respiratory system, viz. T-B, P, and L, should be considered as a composite of mass 1,000 g when calculating "lung dose" (and when applying the weighting factor of 0.12 to lung dose during calculation of the new "effective dose equivalent"). The mathematical procedure used for calculating the ALI values involves calculation of the total activity deposited in the three compartments and divides the total by the combined mass of 1,000 g. The result is numerically the same as calculating a mass-weighted average of the dose to the three compartments.

There is still some controversy among lung dosimetry experts, about whether such an average dose or the dose to the pulmonary compartment alone is the most pertinent. In addition there are still some who feel that the inclusion of the N-P region in the average is appropriate. In light of the current discussion the method of averaging the dose across the four compartments of the respiratory system as used in MILDOS is acceptable.

The dose to the bronchial epithelium from Rn-222 is calculated in a straightforward way, except for a discrepancy in the number of hours assumed for indoor occupancy. The Rn-222 dose calculation assumes indoor occupancy for 24 hours per day while the external dose calculation assumes 14 hours per day are spent indoors. The first value is inconsistent with the second. Regardless of whether or not these assumptions have little effect on the final calculated dose, I recommend that a single value (perhaps 14 hours per day) be adopted for both sets of calculations.

The use of the adult dose conversion factors for calculation of inhalation doses for all ages is acceptable, if the units of the factors used are dose or dose rate per unit concentration in the inhaled air, viz. mrem per pCi m^{-3} of air as given in Table 3, page 31 of RH 802-4 (USNRC, 1979). The lower breathing rate for younger ages compensates for the smaller organ sizes, so that the concentrations of the radionuclides in the organ would be similar for the four age groups,

based on the assumptions that metabolic factors and biological halftimes are relatively independent of age.

3.2.3 External Dose

Except for the inconsistency in hours spent indoors mentioned above, the external dose calculation methods are satisfactory.

3.2.4 Ingestion Doses

With one exception, the method employed in MILDOS for calculating ingestion doses is satisfactory. The exception is the use of total-body dose as a surrogate for lung dose for those nuclides for which no ingestion factors, f_w, are given by ICRP for the lung. Standard practice has been to ignore the lung dose for such nuclides, because if the fraction deposited there were thought to be significant, then the ICRP would have estimated values for f_w in the lung.

The MILDOS procedure overestimates the lung dose (Schermerhorn and Ryan, 1980). The overestimation is especially large for alpha emitters, whose energy does not readily penetrate to adjacent organs. The majority of the effective energy, and hence the dose calculated for a deposited alpha emitter, results from the relatively high energy of the alpha particle ($\sim$5 MeV) multiplied by the quality factor Q. The penetrating gamma radiation makes little contribution to the total-body dose, but would be the only component reaching the lung from an alpha-emitting nuclide deposited in some other organ.(a)

It does not matter that for the nuclides and scenarios considered in MILDOS, the numerical contributions of the pseudo-lung doses from ingestion are numerically small. A mathematical procedure has been promulgated which could lead to improper, large pseudo-lung doses if used for other exposure situations and radionuclides. Therefore, I recommend that no lung doses be calculated for nuclides for which no value of f_w-lung has been set by ICRP. Exceptions would be for nuclides of elements which are essentially uniformly distributed in the body, such as H-3, C-14, Na-22, Na-24, etc.

(a) An example of a radionuclide which does deposit in the lung following ingestion is Cs-137. The lung dose can be readily calculated and can be a measurable addition to the total lung dose from inhalation of Cs-137 in some instances.

3.2.5 Population Doses

Population doses are calculated over a 100-year period, using a procedure analogous to that employed by the Environmental Protection Agency (EPA) when it calculates its Environmental Dose Commitment (EDC)(EPA-1974). Calculation of this type of population dose integrated over long time periods is required if one is to estimate total possible health effects in a population. The choice of 100 years as an integrating period seems rather arbitrary considering the long radioactive half-lives of the nuclides in the uranium decay series. Nevertheless, the U.S. EPA has set a precedent in this matter (EPA-1974), and some justification can be found for continuing to use 100 years. However, the description of the calculation scheme given in Appendix B of RH 802-4 (USNRC, 1979) is not clear. It is possible to derive different dose calculation schemes from the text and Table B-1, page 60. The description should be rewritten to minimize misunderstanding.

MILDOS assumes that a constant annual dose would exist in each of the three operational phases and carries each calculation out to 100 years past exposure so that the cutoff year for each phase's exposure is different.

The dose conversion factors used are those for a 50-year dose commitment from a one-year exposure. Use of such factors for calculation of the EDC introduces conservatism by extending the EDC period to as long as 150 years for exposures in the last year.

3.3 CONCLUSIONS

It is recommended that MILDOS be modified to:

1. Eliminate the calculation of a pseudo-lung dose from ingested radionuclides which has no accepted value for transfer from blood to lung.
2. Use the translocation factor in the manner in which it was intended; viz., apply a value of 1.0 for green leafy vegetables and forage and a value of 0.1 for all other crops.

Adoption of the first recommendation would eliminate an improper procedure that, if applied to other nuclide mixtures, could greatly overestimate the so-called lung dose. In addition, it is recommended that the methodology for calculating the EDC be more fully explained.

One additional minor recommendation is that the number of significant digits attached to the values of poorly known data be reduced to one, or at most, two. This seems appropriate since some values can be estimated to within only an order of magnitude.

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PNL-3772
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Technical Information (3)
Publishing Coordination (2)

NRC FORM 335 (11-81)

U.S. NUCLEAR REGULATORY COMMISSION

BIBLIOGRAPHIC DATA SHEET

1. REPORT NUMBER *(Assigned by DDC)*
NUREG/CR-2022
PNL-3772

4. TITLE AND SUBTITLE *(Add Volume No., if appropriate)*
Technical Review of the Dispersion and Dose Models Used in the MILDOS Computer Program

2. *(Leave blank)*

3. RECIPIENT'S ACCESSION NO.

7. AUTHOR(S)
T.W. Horst, J.K. Soldat, T.J. Bander

5. DATE REPORT COMPLETED
MONTH: March YEAR: 1982

9. PERFORMING ORGANIZATION NAME AND MAILING ADDRESS *(Include Zip Code)*
Pacific Northwest Laboratory
Richland, WA 99352

DATE REPORT ISSUED
MONTH: May YEAR: 1982

6. *(Leave blank)*

8. *(Leave blank)*

12. SPONSORING ORGANIZATION NAME AND MAILING ADDRESS *(Include Zip Code)*
Division of Waste Management
Office of Nuclear Material Safety and Safeguards
U.S. Nuclear Regulatory Commission
Washington, DC 20555

10. PROJECT/TASK/WORK UNIT NO.

11. FIN NO.
FIB B2215

13. TYPE OF REPORT

PERIOD COVERED *(Inclusive dates)*

15. SUPPLEMENTARY NOTES

14. *(Leave blank)*

16. ABSTRACT *(200 words or less)*

This report reviews the technical basis of the models used in the MILDOS computer code. Two major areas are addressed: the models used for atmospheric dispersion, and the models used in the food chain and in human dosimetry.

The atmospheric dispersion review investigates relevant topics, such as diffusion meteorology, plume rise, deposition, and resuspension. The environmental analysis review investigates the food chain model involving retention and translocation assumptions as applicable to the ingestion pathway in humans. In addition, the human dosimetry model used in MILDOS is discussed in terms of all the appropriate potential pathways for human exposure.

Suggested modifications are presented for possible revisions of the MILDOS computer program.

17. KEY WORDS AND DOCUMENT ANALYSIS

17a. DESCRIPTORS

17b. IDENTIFIERS/OPEN-ENDED TERMS

18. AVAILABILITY STATEMENT
Unlimited

19. SECURITY CLASS *(This report)*
Unclassified

21. NO. OF PAGES

20. SECURITY CLASS *(This page)*
Unclassified

22. PRICE
$

NRC FORM 335 (11-81)